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# **Insurance for the future? Potential avian community resilience in cities across Europe**

*Running head: Bird community potential resilience in European cities*

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## Abstract

Urbanization is affecting avian biodiversity across the planet, and potentially increasing species vulnerability to climate. Identifying the resilience of urban bird communities to climate change is critical for making conservation decisions. This study explores the pattern in bird communities across nine European cities and examines the projected impact of climate change in order to detect communities facing a higher risk of functional change in the future.

First, generalized linear mixed models were used to explore the potential resilience of urban bird communities in nine European cities, and the effects of land cover, latitude, abundance of potential domesticated predators (dogs and cats), and bird species richness in each trophic guild. Bird community resilience was represented by an index of functional evenness, because it indicates relatively uniform functional space within the species assemblages. Second, bird community resilience in each city was compared with projected changes in temperature and precipitation for the year 2070 to explore potential future threats to conservation.

The results showed that community resilience was not significantly associated with land use or predator abundance. The number of granivorous and granivorous-insectivorous species increases the potential resilience of the community, while the numbers of insectivores, carnivores and omnivores are negatively correlated with resilience. Of the nine cities, Madrid and Toledo (Spain) are projected to experience the largest change in temperature and precipitation, although their bird communities are characterized by relative high resilience.

In contrast, Rovaniemi (Finland) is projected to experience the second highest increase in temperature and the bird community is characterized by low resilience. These findings indicate the importance of future research on the combined effect of urbanization and climate change on urban biodiversity.

**Keywords:** biotic homogenization; bird diversity; community resilience; conservation; functional evenness; urbanization

## INTRODUCTION

### Global urbanization and ecosystem transformation

We are living in a new self-induced geological era, called "the Anthropocene", characterized by the increasing pressure of urban development on ecosystem dynamics across the entire planet (Crutzen and Stoermer 2000; Lewis and Maslin 2015). The change in land-use produced by the urbanization process is one of the major drivers of environmental modification, with strong and deep impacts on both climate and global biodiversity (Foley et al. 2005; Grimm et al. 2008; Aronson et al. 2014). Expanded urbanization leads to habitat fragmentation and degradation (Spellerberg 1998; Schmiegelow and Mönkkönen 2002; Sklenicka 2016), a process which negatively impacts biodiversity at different levels of organization (Crooks et al. 2004; Wilson et al. 2016). Thus, the mitigation of the loss of biodiversity is partially dependent on our understanding of how urbanization structures biological communities and the subsequent development of wildlife management strategies that incorporate urban ecosystems (Miller and Hobbs 2002).

The effects of urbanization on biodiversity are complex and mainly negative (McKinney 2002; Grimm et al. 2008; Secretariat of the Convention on Biological Diversity 2012; Aronson et al. 2014; Ibáñez-Álamo et al. 2016; Morelli et al. 2016). Especially when trying to assess the effects of urbanization on overall biodiversity, emphasizing that approaches focusing only on species richness are limited by their failure to take the ecological role of species on communities into account (Safi et al. 2013). In fact, estimates of functional diversity often measure variation in ecosystem functioning better than taxonomic diversity measures (Petchey et al. 2004). Among the most recognized effects of urbanization on species assemblage composition stands the 'biotic homogenization' of global communities (Clergeau et al. 2006; McKinney 2006). Biotic homogenization consists of the range expansion of cosmopolitan or generalist species and simultaneously the range contraction of regional and endemic species, a process which is particularly intense in dense urban settlements (Garcillán et al. 2014). Biotic homogenization was confirmed in several taxa (McKinney and Lockwood 1999; La Sorte et al. 2007; Knop 2016), but has so far mostly been studied in urban bird communities (Crooks et al. 2004; Devictor et al. 2007; Godet et al. 2015; Ibáñez-Álamo et al. 2016; Vázquez-Reyes et al. 2017). Urbanized areas (farmlands, villages and cities) can provide additional habitat or food sources, attracting

specific bird species (Evans et al. 2009a; Evans et al. 2009b; Tryjanowski et al. 2015; Reynolds et al. 2017). However, urban bird communities are more homogeneous and often characterized by the absence of specialists when compared with communities from natural habitats (Jokimäki and Kaisanlahti-Jokimäki 2003; Shochat et al. 2010; Ferenc et al. 2014). These differences lead to a gradual decline in the overall functional diversity of species assemblages (Pauw and Louw 2012; Kang et al. 2015; Schütz and Schulze 2015), potentially also reducing the capacity of such communities to resist drastic changes in land use or weather. In fact, it has already been demonstrated that urban development patterns and green areas within cities affect ecosystem dynamics, modifying their abilities to cope with disturbance and modifying their ecological resilience (Alberti and Marzluff 2004).

### **Functional surrogates of potential community resilience**

Ecological resilience is associated with the capacity of ecological systems to resist invasions, climate or land use changes (Haegeman et al. 2016). This term was introduced in ecology in the 1970's by Holling (1973) and has been more recently defined as “the capacity of a given system to change in order to maintain the same identity” (Folke et al. 2010). Thus, resilience is associated with a sort of ‘elasticity’ of the system. This property is associated with a relative functional redundancy of components of that system (Haegeman et al. 2016). Redundant or pseudo-redundant species can be described as species sharing many functional traits, belonging to the same guild or having a similar role in the ecosystem (Gitay et al. 1996). Theoretically, the loss or gain of such species should not strongly affect the overall ecosystem functions (Loreau 2004).

Some studies have suggested that indices of functional diversity such as functional evenness may be useful surrogates for the resilience of communities (Mason et al. 2005; Villéger et al. 2008; Lee and Martin 2017), even if in recent years some potential drawbacks were raised (Ricotta et al. 2014; Legras and Gaertner 2018). These indices could be used to assess the level of utilization of available resources in a given space, by the species inhabiting it (Mason et al. 2005; Mouchet et al. 2010). In communities with high functional evenness the resources would be more efficiently used due to a more uniform distribution of the abundance of species throughout functional space defined by the species traits (Lee and Martin 2017) (see a schematic exemplification in the Fig. S1, ESM). In contrast, in communities with low functional evenness available resources could be underexploited, making the community more susceptible to alterations, as for instance

biological invasions (Elton 1958; Shea and Chesson 2002). More resilient ecological systems should be able to absorb larger shocks, alleviating the effects and reducing the significant modifications, offering a sort of insurance for the future, facing climate change scenarios (Folke et al. 2002). In this context, the characteristics of the species composing each community, as for example the type of diet, acquires significant importance to determine the overall resilience capacities of such a community. We can expect that communities composed of many species which are similar in terms of feeding traits (e.g. several omnivorous species), could be better prepared to face eventual change in land use or climate, because such communities are able to respond better by adapting to fluctuations in food resources. On the other hand, communities composed of several species characterized by a narrow diet (e.g. exclusively frugivorous) could be associated with a higher extinction risk (Terborgh and Winter 1980).

### **Climate change scenarios and the challenge for conservation of biodiversity**

The effects of climate change could be an additional pressure on ecosystems, interacting with land use change and fragmentation (Eglington and Pearce-Higgins 2012), for example by promoting biological invasions (Opdam and Wascher 2004; Bellard et al. 2013). Projections suggest that in the next decades human-induced climate change is expected to continue, and probably accelerate significantly in association with the global emissions of heat-trapping gases (some sources for projections: for America, <https://nca2014.globalchange.gov/>; for worldwide, <https://gisclimatechange.ucar.edu/>) (Hulme et al. 1999; NCAR community 2012; Brown and Caldeira 2017).

There exists a vast amount of scientific literature focusing on the main effects of climate in bird populations, mainly produced by temperature and precipitation (Huntley et al. 2008; Askeyev et al. 2018; Trautmann 2018). The main effects of climate change on birds could be associated with changes in distribution range of avian species, phenology and breeding success, but also genetics and overall population sizes (Trautmann 2018). Briefly, the main effects of temperature can be associated with variation in the body mass (Andrew et al. 2018), avian timing of reproduction (Visser et al. 2009) and reproductive performance of avian species (Conrey et al. 2016). Some studies have shown that extreme temperatures (e.g., heat waves or drastic fall on temperatures) during the early breeding season can negatively affect nesting success of grassland birds (Conrey et al. 2016; Zuckerberg et al. 2018). On the other hand, the main effects of precipitation on birds are associated with the



direct survival of adults and nesting and hatching success. For example, it was demonstrated that large amounts of precipitations during the year preceding a breeding season, can increase the nesting success of some birds (Zuckerberg et al. 2018). However, excessive precipitations during the breeding season can reduce the nesting success (Zuckerberg et al. 2018). Additionally, intense rain events are expected to reduce food availability (e.g. insect resources) or directly foraging efficiency of birds (Siikämäki 1996). Furthermore, many of the effects of weather variables on birds are combined and associated with different ecological levels (species, populations, communities) (Møller et al. 2010; Skagen and Adams 2012; Stephens et al. 2016; Trautmann 2018).

The use of simulated scenarios to forecast the environmental implications of potential climate or land use changes is a recognized tool for the development of ecological policies (Princé et al. 2013). Climate change causes different ecological fingerprints (Parmesan 2006): Bird populations will be affected in many ways, causing ecological adaptations in response to such changes (Butler and Taylor 2005; Barbet-Massin and Jetz 2015; Howard et al. 2015). Among the ecological responses, a potential geographic shift and spatial re-distribution of species assemblages is a critical focus for ecosystem functioning, with deep implications for conservation (Barbet-Massin and Jetz 2015; Princé and Zuckerberg 2015). Induced changes could also be species-specific, altering the composition of bird communities, with potential effects on overall resilience. Additionally, global climate change is expected to strongly affect even the local climate of cities worldwide (Lauwaet et al. 2015), which constitutes important core areas for conservation of many bird species (Ives et al. 2016; Jokimäki et al. 2018). In fact, recent studies have highlighted the important role of cities for global or regional biodiversity conservation, by providing opportunities for settlement of species (Zerbe et al. 2003; Aronson et al. 2014; Møller and Díaz 2017). Cities can be characterized by a mix between global warming and the urban heat island effect (Oke 1973; Kim 1992). Early identification of specific conservation targets where climate can accelerate the already negative impact of urbanization should be a priority in an increasingly urbanizing planet.

The aims of this study were (a) to explore the patterns and geographic variation in resilience of urban bird communities in European cities, (b) to identify urban bird communities more likely to be threatened by climate change according to future projections in temperature and rainfall, and (c) to explore the characteristics of such bird



187 assemblages in terms of species traits (e.g. diet), in order to increase our understanding of  
188 potential conservation concerns, and, therefore, attempts to mitigate the negative effects  
189 of such changes.

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## METHODS

### Study area, environment and collection of bird data

Fieldwork was performed in nine cities, located along a large latitudinal gradient, in six European countries (Fig. 1). This approach, involving different urbanized areas, is particularly suitable for investigating general patterns (Ibáñez-Álamo et al. 2016). In this study were included only urban areas, with multi-story buildings, single family houses, roads and parks. Our classification of environments as urban (percentage of built-up area >50, building density >10/ha and residential human density >10/ha) followed the description made by Marzluff *et al.* (2001). It has been used in many other studies focusing on urban avian ecology (Clergeau et al. 2006; Loss et al. 2009; Møller et al. 2015; Morelli et al. 2016). We collected data on vegetation cover and land use composition within a distance of 50 m from each survey point (Díaz et al. 2013). Land use / cover categories were classified in 6 types: building (which includes residential building, built with infrastructure and processing areas and roads), trees (isolated trees, tree lines and patches), bushes (which includes plants from gardens), grass, bare soil, and water.

Data on bird species were collected during the 2016 breeding season. The surveys were locally adjusted to the start of the breeding season (e.g. early April in southern Spain or late May in northern Finland). Data on bird species were collected by expert ornithologists, following the standardized methodology of point counts randomly selected (Bibby et al. 1992; Voříšek et al. 2010) within each city (ESM, Table S1). All survey points were visited between 06:00 and 10:00 only during favorable weather conditions (i.e., no rain or strong wind). A total of 5 minutes in early spring and 5 minutes during a second visit in late spring allowed for inclusion of both early breeders and late migrants, minimizing issues related to differences in the detectability of bird species (Kéry et al. 2005). During the point counts all individual birds seen or heard within a 50m distance from the observer were recorded, with the only exception being raptors and nocturnal species, because they are subject to a different strategy of survey. The location of each sampling point was recorded with a GPS to ensure that counts were made at the same points during the two surveys. For each sampling point, we also calculated the abundance of mammals as the mean number of dogs and cats seen within 50m during the two 5-minute point counts.

### Surrogate of potential resilience of bird communities

The bird community in each sampling site was defined as the total number of bird species recorded during the two visits. Thus, species richness was expressed as the largest number of bird species of the combined data for the two surveys performed during the breeding season (Magurran 2004).

For each sampling site or bird community, we estimated two functional diversity indices which are traditionally associated with resilience of the community: Functional evenness ('FEve') (Villéger et al. 2008) and the measure 'FeveR' for calculating the functional evenness of a species' assemblage (Ricotta et al. 2014). Both measures are based on a species-trait approach, which focuses on functional aspects of biodiversity (de Bello et al. 2010). Both indices used in this study as surrogate for bird community resilience were calculated using the feeding and breeding avian niche traits provided in Pearman et al. (2014). The bird traits consists of 73 variables describing the niche of each bird species, including (i) body mass, (ii) food type (14 variables), (iii) behavior used for food acquisition (9 variables), (iv) substrate from which food is taken (9 variables), (v) period of day during which a species are active (3 variables), and (vi) used habitats (38 variables) (Pearman et al. 2014). All variables, except for body mass, are binary variables (scored as either 0 or 1) (see a complete description of the used bird traits in ESM, Table S2). The functional traits are associated with many different characteristics of birds including morphological, physiological, and phenological attributes of species, which are related to individual fitness via their effects on growth, reproduction and survival (Violle et al. 2007).

The functional evenness (FEve) indicate how regular is the degree to which the biomass of the species assemblage is distributed in niche space to allow effective utilization of the entire range of resources available (Villéger et al. 2008) and is calculated as follows:

$$FEve = \frac{\sum_{i=1}^{S-1} \min\left(PEW_{i, \frac{1}{S-1}}\right) - \frac{1}{S-1}}{1 - \frac{1}{S-1}}$$

where  $i$  is the species (or functional unit),  $S$  is the total species richness and  $PEW$  is the partial weighted evenness.

The FeveR index reflects the regularity in the distribution of abundances of species, together with the evenness in their pairwise functional dissimilarities (Ricotta et al. 2014), and is calculated as follows:

$$R_U = \frac{E_U - 1/N}{1 - 1/N}$$

where  $E_U$  is the index of Bulla (1994) and  $N$  the number of species, calculated as follows:

$$E_U = \sum_i^N \min\{\pi_i, 1/N\}$$

where  $\pi_i$  is the relative contribution of species  $i$  to  $U$  (average community uniqueness  $U$ ).

The average community uniqueness  $U$  is the expected dissimilarity between one individual of species  $i$  chosen at random from a given community and all other  $j$ -th species in the community, calculated as follows:

$$U = \sum_{j \neq i}^N p_i \times U_i = \sum_i^N p_i \sum_{i \neq j}^N \frac{p_j}{1 - p_i} d_{ij}$$

where  $d_{ij}$  is the functional dissimilarity between species  $i$  and  $j$  (with  $d_{ij} = d_{ji}$  and  $d_{ii} = 0$ ),  $p_i$  and  $p_j$  are the relative abundance of species  $i$  and  $j$  respectively, and  $U_i$  represent the expected dissimilarity between species  $i$  and all other species in the community (Ricotta et al. 2014).

The functional diversity indices used in this study were calculated using the 'FD' package for FEve (Laliberté et al. 2015) and the function provided in Ricotta *et al.* (2014) for FeveR.

Additionally, we calculated species richness for each trophic guild: Granivorous (diet containing primarily seeds and grain), insectivorous (diet containing primarily insects and other invertebrates), granivorous-insectivorous (diet containing primarily seeds, grain, insects and other invertebrates), and carnivorous and omnivorous birds. The trophic guilds were defined by crossing information on bird species from published sources (Cramp and Perrins 1994; Storchová and Hořák 2018) and the 14 food types described in Pearman et al. (2014). With the species richness per each trophic guild we estimated also the diet diversity in each community by applying the Shannon-diversity index (Shannon 1948).

## Climate change projection data: differences in temperature and precipitation

Data on climate change projections were obtained from NCAR GIS Program, through Climate Change Scenarios, version 2.0, 2012. URL: <https://gisclimatechange.ucar.edu/>. Data Access Date: 25/05/2018 (NCAR community 2012). The climate change scenarios have been redesigned for the Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report. The Community Climate System Model (CCSM) as a community-wide effort led by the National Center for Atmospheric Research (NCAR, URL: <https://ncar.ucar.edu/>), and it is a key component of the National Science Foundation program on Climate Modeling, Analysis and Prediction. We used the data provided by CCSM, because they are one of the world's leading general circulation climate models, and a community wide effort led by NCAR. The data define the Representative Concentration Pathways (RCPs), which provide concentrations of atmospheric greenhouse gas (GHG) and the trajectory that is taken over time to reach those concentrations. The spatial resolution of CCSM-3 climate change projections is approximately 1.4 x 1.4 degrees, and represents a plausible alternative scenario for the future, not a prediction or forecast (Moss et al. 2008). We downloaded three projected scenarios for current and future climate change in the years comprised between 2017 and 2070: scenarios RCP 4.5, 6.0 and 8.5. In order to visualize the potential climate change expected in Europe, we used the differences in annual means (raw CCSM data) between 2017 and the simulated data for 2070 in a) temperature (in °Celsius) and b) amount of precipitation (in mm) in each cell provided in the CCSM data.

The three different scenarios were mapped for the European continent (ESM, Fig. S2) by using the ArcGIS 10.1 (ESRI 2012) tool “inverse distance weighted” (IDW). The IDW is a technique that interpolates a raster surface from several point values, taking into account the distances among points (Lu and Wong 2008). The distance among points is uniform across Europe and corresponds to the distance among the centroids of the cells provided in the CCSM data (ESM, Fig. S3). This technique (IDW) is adequate to visualize the areas more subject to potential changes in terms of both temperature and precipitation. We used the maps produced to overlap with the cities where bird community resilience was calculated, and then extracted the relative values of climate change expected for each city. For visualization of data, we set the IDW on 6 different classes from the minimum to the maximum values of delta temperature and precipitation, by using the natural breaks

(Jenks) classification. This classification identifies the best group similar values and maximizes the differences between classes (De Smith et al. 2007).

Finally, in order to compare the congruence/mismatch between potential resilience of urban bird communities in nine European cities and climate change scenarios, we focused on the high pathway in which radiative forcing reaches 8.5 Watts per square meter ( $\text{Wm}^{-2}$ ) by 2100 (Moss et al. 2008), because the worst scenario is the most suitable to better highlight potential areas of largest changes in temperature and precipitation in the continent. The main justification for this choice is supported by a recent study suggesting that the worst predictions of climate change are probably the most accurate (Brown and Caldeira 2017).

## Statistical analyses

In order to investigate the strength and direction of the association between the two indices of functional diversity (FEve and FeveR) we used the Spearman correlation coefficient (Triola 2012). Because both indices were significantly positively correlated ( $R^2 = 0.55$ ,  $p = 4.6\text{e-}12$ ), the subsequent analyses only focused on one of these indices. We chose FEve because this variable showed a largest range if compared with FeveR in each city, guarantying a better visualization of data. A linear regression was used to explore the potential association between the indices of functional diversity and bird diet diversity in avian communities.

To explore differences in avian niche traits among species of different trophic guilds, the analysis of dissimilarity was used. In this procedure, a dissimilarity matrix was constructed by the “Gower” dissimilarity measure among bird species, considering the 73 avian traits described above. We used the function ‘daisy’ from the package ‘cluster’ in R (Maechler et al. 2018). The average dissimilarity with all the 279 species in the pool was calculated for each bird species. The Kruskal-Wallis and post hoc Kruskal-Dunn were used to check for significant differences in species dissimilarities among the five trophic guilds. The post hoc contrasts were performed using the package ‘PMCMR’ in R (Pohlert 2014).

We used Mantel tests to check for spatial autocorrelation of data (SAC) (Mantel 1967), based on Monte Carlo permutations with 9999 randomizations to test for significance (Oksanen et al. 2016). Sampling sites were treated as statistically dependent observations because the values of SAC between geographic distance (km) and FEve distance among

336 sites was very low though statistically significant ( $r_M = 0.048$ ,  $n = 588$ ,  $p = 0.001$ ) (Manly  
337 2006). In order to alleviate SAC issues, the geographical coordinates of sampling sites  
338 were introduced as covariates to incorporate spatial variation during the modeling  
339 procedure (Legendre 1993; Dormann et al. 2007).

340 Generalized Linear Mixed Models (GLMMs) were used to study the patterns of bird  
341 community resilience (FEve) in relation to geographical coordinates of sampling sites, land  
342 use / cover composition around the point count, abundance of mammals (number of dogs  
343 and cats) and bird species richness for each trophic guild (granivorous, granivorous-  
344 insectivorous, insectivorous, carnivorous and omnivorous), modeled as fixed effects. In  
345 order to avoid any redundancy in the modeling procedure, we checked for the potential  
346 association between the index of FEve (which was estimated considering habitat, breeding  
347 and feeding traits) and the Shannon index estimated based on the species richness for  
348 each trophic guild by using a simple linear regression analysis. The association between  
349 these two variables was not statistically significant (ESM, Fig. S4;  $F = 3.2$ ,  $df = 1$ ,  $586$ ,  $p >$   
350  $0.05$ ), justifying the use of both during the modelling procedure. The following predictor  
351 variables were too strongly correlated and thus were removed, in order to avoid  
352 multicollinearity issues (Graham 2003): building (negatively and significantly correlated  
353 with grass and tree), bare soil (negatively and significantly correlated to grass) and overall  
354 species richness (positively correlated to species richness for each trophic guild). City was  
355 included as a random effect to account for possible consistent differences among cities.  
356 Models were fitted by maximum likelihood, using the package 'lme4' in R (Bates et al.  
357 2014). The model selection was based on Akaike's Information Criterion (AIC) (Burnham  
358 and Anderson 2002) with the package 'AICcmodavg' in R (Mazerolle 2016).

359 All statistical tests were performed with R software version 3.2.4 (R Development Core  
360 Team 2019).



## RESULTS

### Bird community resilience in nine European cities

A total of 107 bird species were recorded at 588 point-counts distributed in nine European cities (ESM, Table S1). The complete list of species is provided in the Electronic Supplementary Material (ESM, Table S3). Species richness per point count ranged from a minimum mean of 7.52 species (min: 3, max: 12 species) in Munich to a maximum mean of 14.14 species (min: 3, max: 30 species) in Granada.

Overall, the two surrogates of bird community resilience varied between a minimum of 0.69 (FEve) and 0.80 (FeveR), and a maximum of 1.00 (FEve) and 0.99 (FeveR). The differences in FEve among cities were statistically significant (ANOVA:  $F = 51.08$ ,  $df = 8$ ,  $579$ ,  $P < 2e-16$ ). The lower mean value of bird community resilience (FEve) was found in Munich (0.87, min: 0.69, max: 0.95), while higher mean values were found in the Spanish cities of Madrid (mean: 0.95, min: 0.90, max: 1.00), Granada (mean: 0.94, min: 0.86, max: 0.99) and Toledo (mean: 0.94, min: 0.88, max: 0.98) as well as in the city of Prague (mean: 0.94, min: 0.85, max: 0.98) (Fig. 1).

The values of bird community resilience (FEve) in nine European cities were not significantly correlated with the environmental characteristics around the point counts (land use / cover composition) nor with the abundance of dogs or cats (Table 1). Additionally, we did not find evidence of latitudinal or longitudinal effects on resilience (Table 1, Fig. 1). Bird community resilience was higher in cities where the relative number of granivorous or granivorous-insectivorous species was higher, while it was negatively associated with the richness of insectivorous, carnivorous and omnivorous-scavenger bird species (Table 1, Fig. 2).

Overall, avian trait dissimilarity was higher in carnivorous and omnivorous species than in granivorous, granivorous-insectivorous and insectivorous birds. These differences were statistically significant (Kruskal-Wallis  $\chi^2 = 43.98$ ,  $df = 4$ ,  $P = 2e-09$ , ESM, Fig. S5).

### Avian resilience and climate change scenarios

Considering the projection of climate change scenarios for Europe in the next fifty years, provided by the Intergovernmental Panel on Climate Change (IPCC) in the Fifth Assessment Report, the cities exposed to a larger change in temperatures (higher than

2.4°C) are the Spanish cities of Madrid, Granada and Toledo and the Finnish cities of Rovaniemi and Turku (Fig. 3). On the other hand, the European cities more exposed to changes in terms of average precipitation in mm are Madrid, Groningen and Poznan (Fig. 3). Overall, the cities less exposed to a drastic variation in terms of temperature and precipitation under a high pathway climate change projection are cities from central Europe such as Prague and Munich (Fig. 3, ESM, Table S4).

Crossing the mean values of bird community resilience for the nine European cities with the values of expected delta temperature and precipitation, explored in a 3D plot, highlight some potential scenarios of urban bird conservation for the next fifty years: Even if Munich was the city with the lowest avian community resilience, it seems however that bird communities within that city are not affected by strong climate change events, but only by a relative moderate change in temperature, with temperature increases of 1.9-2.3 °C, but less strong changes in precipitation when compared with other European cities (ESM, Fig. S6). The Spanish cities with higher values of bird community resilience will face more notorious climate change events: Madrid (with the higher bird community resilience) could be affected by a relatively large change in the mean amount of monthly precipitation (110-170 mm) as well as increasing temperatures around 2.4-2.8 °C (ESM, Fig. S6). Granada and Toledo could face similar scenarios of increasing temperatures, but less strong changes in terms of mean amount of monthly precipitation (ESM, Fig. S6). The city of Prague was characterized by bird communities with relative high resilience, and simultaneously not exposed to so drastic climate change events in the next fifty years (ESM, Fig. S6). The Finnish city of Rovaniemi could suffer an important increase in average temperatures, while their bird species assemblages are characterized by lower resilience than other urban bird communities, suggesting a potential conservation problem.

## DISCUSSION

### Climate change scenarios in cities and bird community resilience

In this study, we suggested potential future conservation concerns in urban bird communities, by crossing information on a surrogate of species assemblage resilience (FEve) (Mouchet et al. 2010) and the projection of scenarios facing climate change. Even if we examined three different scenarios, we used the high pathway climate change scenario because it could be the most plausible (Brown and Caldeira 2017).

Our findings provide important information about the cities that could face stronger climate change events, and the capacity of their bird communities to reduce any negative impact associated with them. Briefly, a high probability of climate change and simultaneously a low value of bird community resilience can be interpreted as the worst scenario, suggesting a potential conservation concern.

A higher pressure of climate change in terms of delta temperatures and delta precipitation should mainly affect cities from the Southern and Northern regions of Europe. The data derived from the projections made by NCAR Climate Change Scenarios (NCAR community 2012) showed that cities like Granada, Madrid and Toledo in Spain and Rovaniemi and Turku in Finland could suffer variation in temperatures larger than 2.6°C before the year 2070 (ESM, Table S4). Additionally, Madrid and Toledo could also be exposed to a significant change in the amount of monthly precipitations. Finally, Madrid was also the biggest city surveyed in this study, with more than three million people (ESM, Table S1). This fact is important, if we consider that the number of threatened bird species could be positively associated with human population size (Pautasso and Dinetti 2009). However, these two Spanish cities are characterized by species assemblages with higher resilience, a fact that can offer insurance against the ability of communities to respond to challenges posed by climate change. On the other hand, from the two Finnish cities, Rovaniemi could constitute a potential problem for conservation in the future: This northern city could face an important increase in average temperatures, while their bird communities are mostly characterized by lower values of resilience within the cities used in our study. Especially in the case of Rovaniemi, most species are (long)-distance migrants (Schaefer et al. 2008; Saino et al. 2011) and in most cases insectivores (Fig. 2).

Even if all cities could be exposed to potential climate change scenarios, the cities from the central part of the continent (Groningen, Munich, Poznan and Prague) are less exposed to drastic changes in temperatures, when compared with the other cities studied (Fig. 3), suggesting a low conservation concern of urban bird communities. This is important when planning future conservation, regarding both urbanization and climate change. For example, Munich was the city with the lowest mean value of resilience of avian species assemblages, but considering that future scenarios suggest a relative small impact due to climate change, we consider that specific conservation actions attempting to promote resilience of urban bird assemblages in this city are not urgently needed. In contrast, other cities with low/medium levels of resilience, like Rovaniemi, which will be exposed to important increases in temperature in the future, should be the focus of conservation practices or at least close monitoring schemes. The bird communities from Munich were characterized by a high proportion of insectivores and simultaneously a low proportion of granivorous species, when compared with the other European cities. When comparing Munich with the nearby city of Prague, Czech Republic, we can highlight how the risk of increase in the level of threat for bird communities in Prague could be lower, because the city presents species assemblages with high values of potential resilience, also being a city not subject to particular intensification of climate change.

Another important result to highlight from this study is the weak correlation between potential resilience of urban bird communities and land use / cover composition where the communities were assessed. This lack of association in our study may suggest that little variation in land use / cover composition or heterogeneity in urban green areas do not have a strong impact on the regularity of bird traits in functional space and how efficiently the resources are utilized. A study focused on anthropized environments (farmlands) showed how functional evenness of bird communities is only weakly correlated with the environmental characteristics of the area (green area size, canopy heterogeneity) (Lee and Martin 2017).

We found in the nine European cities that urban bird communities with high richness of granivorous or granivorous-insectivorous species were the communities with higher potential resilience. In contrast, urban bird communities with an overrated number of insectivorous, carnivorous and omnivorous species were assemblages with lower potential resilience. The main reason why carnivorous and omnivorous species richness was

negatively associated with potential community resilience could be explained by the fact that those bird species were characterized by higher trait dissimilarity, when compared with the entire pool of species (ESM, Fig. S5). The granivorous birds recorded in this study, on the other hand, were more similar in terms of breeding and feeding traits (ESM, Fig. S5). By increasing the number of species which are functionally closer, the functional redundancy will rise, achieving a higher functional evenness and potential bird community resilience.

Regarding the bird's foraging characteristics in relation to the potential resilience of species assemblages, however, we consider it relevant to highlight the limitations of simple traits describing the diet of a species. Further studies should also consider the fact that many species are relatively plastic in terms of foraging ecology. Some species, as shrikes or sparrows, can exploit different types of diet depending on the geographical context and environmental conditions, being also scavengers at roadsides (Tryjanowski et al. 2003; Morelli et al. 2015). The diet plasticity of bird species, also in cities, can shift their main type of food between seasons. E.g. in the northern part of Europe, while most species need/use protein-rich arthropods during breeding season for their nestlings, they must change their diet for berries during winter, because of a lack of insects (e.g. Turdidae, Sylvidae). Geographic variation in trophic plasticity of bird species could play an important role in the overall potential resilience of the communities. Since specialist species are considered more prone to the processes that lead to extinction than generalist species (McKinney 1997; Colles et al. 2009), we can expect that a given community composed of many species characterized by high ecological plasticity in terms of foraging (but also behaviour or breeding) should be better adapted to eventual environmental or climatic changes, because such species could respond better by adapting to different types of food or other resources. Further studies could also focus on the overall level of specialization of bird communities inhabiting cities (Morelli et al. 2019).

Finally, the fact that we did not find any evidence of latitudinal or longitudinal effects on avian resilience among the nine cities could be interpreted as a confirmation of the level of homogenization of urban settlements in Europe (Devictor et al. 2007).

## **Final considerations and some remarks on the use of a surrogate of community resilience**

Recent studies have provided evidence of drawbacks for the use of this functional diversity index (e.g. Ricotta et al. 2014; Legras and Gaertner 2018), which is the evenness component of functional diversity. Briefly, the main concerns are related to the fact that an increase in FEve index values is not always synonymous with an increase in functional evenness, mainly when comparing communities which differ in terms of abundance distribution (Legras and Gaertner 2018). However, in the present study, we estimated the FEve crossing a trait matrix with a matrix of solely presence/absence of species, therefore overcoming (or at least alleviating) this potential problem. Additionally, we explicitly compared the behavior of the index FEve with the new index proposed by Ricotta et al. (2014), which measures the regularity in the distribution and abundance of species in functional space, together with the evenness in their pairwise functional dissimilarities, to ensure that in our study both could be used similarly. Moreover, any index or metric used to quantify the hypothetical “resilience” of a species assemblage must be handled cautiously. An index is not a direct "measure", especially because the resilience or capacity to respond to an alteration depends on many (and complex) factors: period of disturbance, intensity, relative plasticity of species, biotic interactions (explicit and hidden), and cascade effects (Spears et al. 2015; Morelli and Tryjanowski 2016). Ecological resilience was defined as a multifaceted concept (Cumming et al. 2005). Thus, we consider that any index should be more efficiently used when applied mainly to comparisons among sites or areas.

Importantly, climate change effects can interact with the distribution of bird species, facilitating the expansion of invasive birds that could compete, displace and / or prey on native species (Bellard et al. 2013). Therefore, this could potentially amplify the effects of changes in land use, precipitation or temperature. Maintenance of the level of functional redundancy in bird communities can increase the overall tolerance of such species assemblages to potential scenarios of land use and climate change (Elmqvist et al. 2003). Thus, the capacity of species assemblages to face and recover from extreme events (such as climate or land use change) will determine their persistence.

In conclusion, our findings highlighted how an approach combining projections of climate change scenarios and potential resilience of species assemblages (using species trait-based methods), could be useful to identify in advance conservation concerns. We hypothesize that the approach used in this study could also be applied to other taxa such

542 as insects and mammals. This may help establish adequate urban planning strategies for  
1  
543 the promotion of a high diversity of communities in urban exploiter and tolerant species,  
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544 increasing the level of protection of urban ecosystem functioning.  
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